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Interpretation of variations in fine, coarse and black smoke particulate matter concentrations in a northern European city

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Abstract

The $PM_{2.5}$, PM_{coarse} and Black Smoke (BS) particle metrics broadly reflect different source contributions to PM_{10}. The aim of this study was to generate data for $PM_{2.5}$ at an urban background site in the UK, and to use the daily collocated measurement of $PM_{2.5}$, PM_{10} (and hence PM_{coarse}) and BS to yield insight into source influences on particulate matter for input to developing PM air quality policy. Mean daily PM_{10}, $PM_{2.5}$ and BS for a year of measurement in Edinburgh were 15.5, 8.5 and 6.6 μg m^{-3}. The $PM_{2.5}$ data were well-within possible future limit values proposed by the European Commission Clean Air For Europe programme. Daily $PM_{2.5}$ and PM_{10} were significantly correlated (r^2 = 0.75) with $PM_{2.5}$ contributing 54 %, on average, to PM_{10}. The daily BS:PM_{10} and BS:$PM_{2.5}$ ratios were more variable, and significantly lower in summer than in winter, reflecting the greater contribution of non-black photochemical secondary particles to PM_{10} in summer. Analysis with respect to wind showed a dominant influence of dispersion on BS and $PM_{2.5}$ but both a dispersion and a wind-driven suspension influence on PM_{coarse}. The latter was higher than in central England (averaging about one-third of the PM_{coarse}), and greater for on-shore wind direction, suggesting a sea-salt source for this component in addition to other particle resuspension contributions. Overall, the data showed that excursions in PM_{10} were driven more by variations in $PM_{2.5}$ than by PM_{coarse} or BS. Both $PM_{2.5}$ and its proportion to PM_{10} were significantly elevated for air-masses passing over continental Europe and the British Isles, whereas BS varied less with air-mass origin, supporting the conclusion that concentrations of particulate matter, particularly of finer PM, are strongly influenced by regional scale synoptic meteorology (presumed to be predominantly secondary PM), whereas BS is dominated more by local sources. Comparison of BS with a nearby rural site suggested that approximately three-quarters, on average, of the urban BS was local in origin.

Introduction

The concentration of outdoor airborne particulate matter is nowadays principally quantified using the PM_{10} metric. Thus, the European first Daughter Directive (99/30/EC), as for legislation elsewhere, sets limits for PM_{10} in recognition of the effects of particles on human health. PM_{10} is of course not a single entity, but comprises a wide range of particle sizes and chemical composition (as a consequence of the multitude of sources and dispersal processes), and it is becoming common to sub-divide PM_{10} into fine ($PM_{2.5}$) and coarse ($PM_{10-2.5}$ or PM_{coarse}) size ranges. Using this dichotomy it is possible to make the crude generalisation that $PM_{2.5}$ (which is derived mainly from gas-to-particle reactions in combustion exhaust or between ammonia and sulphate and nitrate) is predominantly of direct anthropogenic origin, while PM_{coarse} (which is derived mainly from mechanical suspension of soil, dust, sea-salt and diffuse industrial/traffic-related sources) is predominantly of natural or indirect anthropogenic origin (APEG, 1999; AQEG, 2005).

The increasing availability of $PM_{2.5}$ data for epidemiological studies, principally in the USA, has focused attention on the extent to which this finer fraction of airborne particles is responsible for the health effects previously ascribed to PM_{10}. Following advice from the World Health Organisation (WHO, 2003; WHO, 2004), the Clean Air for Europe Working Group on Particulate Matter recently recommended that $PM_{2.5}$ rather than PM_{10} should become the principal metric for assessing exposure to PM in Europe (CAFE, 2004). However, the CAFE working group has not yet recommended specific limit values for $PM_{2.5}$, stating that more European data are required for $PM_{2.5}$, and for the factors determining relationships between $PM_{2.5}$ and PM_{10} (or PM_{coarse}), in order to derive an appropriate legislative framework for the protection of human health against PM. A summary of available $PM_{2.5}$ monitoring data in Europe has recently been published (Van Dingenen *et al.*, 2004), but caution is required in deriving comparative conclusions because of the variety of sampling and analytical techniques used across the different locations.

An alternative, historic measure of airborne PM is black smoke (BS). This is effectively a measurement of the optical absorption (or blackness) of the PM and can be converted into a notional mass concentration using a standard equation (DETR, 1999). Although the BS sampler size-selects at approximately PM_4 (McFarland *et al.*, 1982), the optical darkness is dominated by the contribution from sub-μm diameter elemental carbon particles (Edwards *et al.*, 1983; Horvath, 1996). Therefore, in the modern-day context, BS is a good surrogate for primary combustion particles, and is largely insensitive to secondary inorganic aerosol and PM_{coarse}. This fact, in conjunction with source-oriented epidemiological studies that show associations between adverse health and traffic- and combustion-derived air pollution (Laden *et al.*, 2000; Roemer and van Wijnen, 2001; Hoek *et al.*, 2002), has led WHO also to recommend re-evaluation of BS as a particle pollution metric (WHO, 2003), in parallel to their recommendation on $PM_{2.5}$.

The paragraphs above show that $PM_{2.5}$, PM_{coarse} and BS broadly reflect different source contributions to PM_{10}. The aim of this study was therefore to analyse a year of collocated daily measurements of BS, $PM_{2.5}$, and PM_{10} (and hence PM_{coarse} by difference) in order to evaluate the factors influencing the daily relationships between these metrics in an urban area in the UK. The data and inferences will contribute to the ongoing development of policy on airborne PM, and in the interpretation of epidemiologic data associating PM air pollution and adverse health effects.

Experimental

Sampling was conducted on the roof of Old College, an urban background site, in central Edinburgh (3° 12′ W, 55° 57′ N) for one year from September 1999. The city of Edinburgh (population ~500,000) is situated near the east coast of Scotland, and has comparatively little heavy industry.

Samples of PM_{10} and $PM_{2.5}$ were collected daily (midnight to midnight) using R&P Partisol 2025 samplers, both with PM_{10} heads and one with a sharp cut cyclone to select further for $PM_{2.5}$. Samples for BS were collected using a standard UK Black Smoke network sampler. Hourly wind velocity data were logged from a collocated RM Young wind vane. A second Black Smoke sampler was located in a portacabin in a field adjacent to the Centre for Ecology and Hydrology, a rural site 17 km SSW of Edinburgh. The Partisols and BS sampler were operated in accordance with guidance specified by the USEPA (1998) and the UK national smoke network (DETR, 1999), respectively. Every 4 weeks the volumetric flow rate of the Partisols was calibrated against a NIST traceable critical orifice flow meter (Chinook Engineering). Trials of the two Partisol instruments operated both as PM_{10} samplers for 14 days, or both as $PM_{2.5}$ samplers for 14 days, yielded correlation coefficients between sample masses of 0.991 and 0.985, respectively.

The Gelman Zefluor filters used in the Partisol samplers were conditioned for 24 h in the weighing room at 17 ± 3 °C and 53 ± 7% RH pre- and post-exposure before being passed under an anti-static ionising blower and triple-weighed on a Sartorius MC5 six-place balance. A set of six weigh-blank filters was interspersed within the pre- and post-weight sessions of each batch of sample filters and the mean change in weigh-blank filter mass between weighing sessions used to correct the sample filter mass changes. The mean weigh-blank filter correction corresponded to only 1.1% and 2.1%, respectively, of the mean PM_{10} and $PM_{2.5}$ sample masses. Green *et al.* (2001) also concluded that there was no significant variability in Partisol sample filters for RH in the range 40 to 60% and temperature in the range 15 to 25 °C.

The reflectance of the Whatman no. 1 filters used in the BS samplers was measured using an EEL Model 43D reflectometer and converted to a mass concentration using the UK BS calibration curve (DETR, 1999). Although this conversion is unlikely accurately to represent true mass concentration of dark particles, it nevertheless provides a relative measure of their concentration.

Air-mass back-trajectories for a mid-day arrival at the 930 hPa pressure level at the sampling site were calculated using the 3-D model available at the British Atmospheric Data Centre. These were categorised by statistical hierarchical clustering using mean Euclidian distance squared and average linkage clustering on the vectors of the 30 variables: distance north of Edinburgh, distance south of Edinburgh, and pressure level, at 12, 24, 36,…120 h prior to arrival. The "best" number of clusters into which to classify the trajectories was adjudged from step-changes in the values of RMS and r^2 plotted against number of clusters (Cape *et al.*, 2000), and examination of the robustness of allocation of individual trajectories to a given cluster. Five major clusters emerged (amalgamating the two Arctic clusters of Heal et al. (2005)), together accounting for >94 % of the total number of trajectories.

Results and discussion

Concentrations

Mean daily PM_{10}, $PM_{2.5}$ and BS for the year of collocated urban measurement were 15.5, 8.5 and 6.6 μg m^{-3}, respectively. Mean daily BS at the rural site was 1.8 μg m^{-3}. The EU Stage 1 limit values (99/30/EC) for PM_{10}, incorporated as statutory Air Quality Standard (AQS) objectives in the UK, are an annual mean of 40 μg m^{-3} and no more than 35 days per year (approximately equivalent to 90th percentile) of daily means exceeding 50 μg m^{-3}. Proposed tighter AQS objectives (to be achieved by the end of 2010) are no more than 7 exceedences per year (~98th percentile) of a daily mean of 50 μg m^{-3} and an annual mean of 20 μg m^{-3} (18 μg m^{-3} in Scotland). In this study there were no days on which daily PM_{10} exceeded 50 μg m^{-3}. This, and the annual mean of 15.5 μg m^{-3}, show that background locations in Edinburgh are already in compliance with the 2010 PM_{10} objectives.

PM_{10} was also recorded over the same period by Tapered Element Oscillating Microbalance (TEOM) at an urban centre location in central Edinburgh, ~0.7 km from the Old College site. Mean PM_{10} recorded at the TEOM site over the same period was 17.8 μg m^{-3}, also with no daily value exceeding 50 μg m^{-3}. Applying the current "gravimetric equivalent" correction factor of 1.3 to the data to allow for loss of semi-volatile PM in the heated inlet of the TEOM (AQEG, 2005) yielded four days with PM_{10} >50 μg m^{-3}, and an annual average of 23.1 μg m^{-3}. The latter exceeded the proposed 2010 annual AQS, indicating it may be difficult to achieve this more stringent AQS for urban centre locations as well as at specific roadside hotspots, even in a relatively unpolluted city such as Edinburgh.

Correlation of daily PM_{10} at the two locations in Edinburgh was fairly high (r^2 = 0.53) suggesting that a significant proportion of variability in (non-roadside) urban PM_{10} is determined by regional-scale, or background, sources and synoptic meteorology. High local windspeed was the only parameter associated with lack of spatial correlation which is probably due to different characteristics of very localised wind-induced particle suspension in the vicinity of each monitor.

There are no AQSs for $PM_{2.5}$ in the UK or elsewhere in Europe. The CAFE Working Group on Particulate Matter suggested 35 μg m^{-3} as a 90th percentile daily limit value, and between 12-20

$\mu g\ m^{-3}$ as an annual value as starting points for further discussion (CAFE, 2004). For the measurements undertaken here, the 90th (and 98th) percentile of daily mean $PM_{2.5}$ was 15.3 (21.1) $\mu g\ m^{-3}$, which together with the annual average 8.5 $\mu g\ m^{-3}$, indicate that background air in Edinburgh would also be in compliance with potential future $PM_{2.5}$ legislation. The annual mean $PM_{2.5}$ in Edinburgh was considerably lower than the $PM_{2.5}$ annual means of >15 $\mu g\ m^{-3}$ reported for six other urban background locations in Europe, but are comparable to those summarised for the rural sites investigated (Van Dingenen *et al.*, 2004).

Inferences of sources from relationships between PM metrics

Daily PM_{10} and $PM_{2.5}$ were strongly correlated ($r^2 = 0.75$, $n = 349$), Fig. 1, with linear regression equation, $PM_{2.5} = 0.61PM_{10} - 0.98$. This is consistent with the strong relationships between hourly $PM_{2.5}$ and PM_{10} from the previous analysis of collocated $PM_{2.5}$ and PM_{10} in the UK (Harrison *et al.*, 2001) which reported r^2 values from 0.60-0.96 and regression coefficients from 0.50-0.80. The recent summary of other European data also reports strong within-site correlation of $PM_{2.5}$ and PM_{10} (Van Dingenen *et al.*, 2004). Since $PM_{2.5}$ is a subset of PM_{10} some correlation is inevitable, but the extent to which they are correlated is relevant to an assessment of the usefulness of designating an additional standard for $PM_{2.5}$.

The distribution of daily ratios of $PM_{2.5}$:PM_{10}, BS:PM_{10} and BS:$PM_{2.5}$, divided into summer and winter, are shown in Fig. 2. The median daily ratio of $PM_{2.5}$:PM_{10} for the year was 0.52 (Inter-Quartile Range, IQR: 0.44-0.62). The narrow IQR is a consequence of the fairly strong correlation between $PM_{2.5}$ and PM_{10}. The average proportion of $PM_{2.5}$ in PM_{10} in Edinburgh (~0.55) is towards the low end of the ranges summarised by Harrison *et al.* (2001) for sites in central and southern England and by Van Dingenen *et al.* (2004) for a range of sites across Europe, although comparable to the proportion reported by Brook *et al.* (1997) for sites across Canada. Edinburgh is a comparatively unpolluted city with a windy coastal location, so is likely to be proportionally less influenced by anthropogenic fine particles and more influenced by suspension of coarser dust and sea-salt.

The median (and IQRs) for daily ratios of BS:PM_{10} and BS:$PM_{2.5}$ were 0.42 (0.27-0.59) (r^2 = 0.18, n = 349) and 0.80 (0.51-1.09) (r^2 = 0.27, n = 362), respectively. The poorer correlation and much wider variability in these latter ratios compared with $PM_{2.5}$:PM_{10} data reflects the fact that black smoke is a strong indicator for a specific source contributor to PM (combustion-derived dark particles) rather than being a measure of all particles in a given size fraction. Fig. 2 also shows that, on average, the ratios BS:PM_{10}, and BS:$PM_{2.5}$, were significantly lower ($P < 0.001$) in summer than in winter, reflecting either or both greater contribution of non-black photochemical secondary particles to PM in summer and increased BS from combustion heating in winter. There are no local sources of BS at the rural site, so from the average rural:urban BS ratio of ~0.25 it can be inferred that nearly three-quarters of BS in Edinburgh is locally generated and the remainder is "background." (The above analysis excluded days when winds were from the N-NE for which Edinburgh is immediately upwind of the rural site. A dispersion source from the city was very clear on these occasions, with rural:urban BS ratios exceeding 0.5).

Fig. 1 reveals slight upward curvature with increasing particle concentration. There was a significant trend for the contribution of $PM_{2.5}$ to PM_{10} to increase with increasing $PM_{2.5}$ (Fig. 3) and, less strongly, with increasing PM_{10}. In addition there was greater variation in daily $PM_{2.5}$ than in PM_{coarse} (90%ile-median values of 8.0 and 5.4 μg m^{-3} for $PM_{2.5}$ and PM_{coarse}, respectively) and little correlation between $PM_{2.5}$ and PM_{coarse} ($r^2 = 0.10$). All this can be interpreted as showing that days of high $PM_{2.5}$ were not also strongly associated with high PM_{coarse} and that variability in daily PM_{10} was more driven by variations in $PM_{2.5}$ than in PM_{coarse}. Van Dingenen *et a*l. (2004) also report a tendency for $PM_{2.5}$:PM_{10} ratio to be lower at sites with lower PM_{10} concentration, except at heavily trafficked kerbside sites where traffic-induced suspension of coarse particles make a substantial contribution to local PM_{10}.

Further insight is gained from analysis of PM size fractions with local windspeed. PM_{10} and BS are plotted against daily average windspeed in Fig. 4 and show clearly distinct relationships. The relationship of PM_{10} with windspeed is scattered but has a U-shape showing levels of PM_{10} to be enhanced at both high and low windspeeds. The observation is consistent with the action of the opposing processes of entrainment of PM_{10} (or components of PM_{10}) with increasing windspeed on the one hand, and dilution by wind on the other. The latter process is illustrated well in Fig. 4 by the strongly monotonic inverse relationship between windspeed and BS, which consists almost entirely of fine particles emitted from local combustion sources. The $PM_{2.5}$ data has a similar relationship to windspeed as BS, indicating that it is the PM_{coarse} component of PM_{10} which includes a windspeed-dependent resuspension source. This source will be wind-driven entrainment of dust on soils and other surfaces. For a coastal location like Edinburgh, this component is likely also to include sea-salt, whose contribution to atmospheric PM will increase with increasing windspeed over the sea surface.

The effect of dispersion dilution on PM_{coarse} can be factored out by normalising with respect to the BS data, as shown in Fig. 5. Despite the scatter in the data there is a significant trend for PM_{coarse}/BS to increase with windspeed, and more strongly in the summer than in the winter. The seasonal difference may be explained by generally drier conditions in summer leading to more facile suspension of particles from the surface, although closer inspection of Fig. 5 shows that highest windspeeds more often occurred in winter.

Using the methodology described by Harrison *et al.* (2001), power-law relationships between windspeed and resuspended PM_{coarse} were derived for each season, from which the overall magnitude contributed by resuspended PM_{coarse} could be estimated. A threshold windspeed for resuspension of 2 m s^{-1} best fit both the summer and winter data. During summer, resuspended PM_{coarse} contributed 1.9 μg m^{-3} (or 26 %) on average to mean total PM_{coarse} (7.1 μg m^{-3}), whilst in winter its estimated contribution was 2.6 μg m^{-3} (or 37 %). Thus, while the data show that resuspension is more facile in summer, the higher average windspeed during winter actually results in a larger component of resuspended PM_{coarse} in PM_{10} during winter. The proportions of resuspended PM_{coarse} derived here for Edinburgh are generally higher than those determined by Harrison *et al.* (2001) for Birmingham, central London and Harwell because of Edinburgh's windier and near-coastal location. This more detailed analysis concurs with the earlier observation that PM_{coarse} comprises a greater proportion of PM_{10}, on average, in Edinburgh, than at the sites investigated by Harrison *et al.* (2001).

The remaining non-wind-suspended component of PM_{coarse} will include primary emissions of coarse particles from industry and construction activity, as well as the important source of traffic-related coarse particles arising via mechanical abrasion from the vehicle and entrainment from the road surface by vehicle-induced turbulence. Analysis of chemical composition data has also demonstrated that non-exhaust vehicle emissions make an important contribution to urban PM in the UK (Harrison *et al.*, 2004; Heal *et al.*, 2005).

A contribution to PM_{coarse} from sea-salt is supported by the wind roses plotted in Fig. 6 which show that PM_{coarse} is elevated when local wind direction is from the north-east and east. This is the coastal sector from central Edinburgh, and a windspeed wind rose (not shown) shows that windspeeds from this sector are higher than average. Fig. 6 also shows that $PM_{2.5}$ is proportionally more enhanced than PM_{coarse} when wind is from the east and this must be due to longer-range transport of particles from continental Europe. The significant influence of long-range transport on concentrations of PM_{10} is well-known (King and Dorling, 1997; Malcolm *et al.*, 2000; Buchanan *et al.*, 2002). Here the influence of long-range transport on different particle measures was investigated according to the trajectory taken by the air-mass arriving at the measurement site. Fig. 7 shows that median PM_{10} on days when air-mass originated from continental Europe or from local circulation about the British Isles (20 and 16 $\mu g\ m^{-3}$, respectively) were significantly elevated compared with days when air-mass originated from Atlantic SW, Atlantic W and the Arctic (12, 13 and 14 $\mu g\ m^{-3}$, respectively). The pattern was accentuated for $PM_{2.5}$ with median concentrations of 11 and 9 $\mu g\ m^{-3}$ for days with air-masses originating from continental Europe or local circulation (i.e. over land) compared with 6 $\mu g\ m^{-3}$ when air-masses originated from the Atlantic and Arctic. Thus change in air-mass source was associated with increases of 50 % or more, on average, of receptor $PM_{2.5}$ in Edinburgh. The ratio of $PM_{2.5}/PM_{10}$ was also significantly elevated on the days with European and local circulation air-masses as shown by the numbers at the bottom of Fig. 7.

In contrast, BS in Edinburgh varied much less with air-mass source than PM_{10} or $PM_{2.5}$. The max/min of the median trajectory cluster values shown in Fig. 7 are 1.7 and 1.9 for PM_{10} and $PM_{2.5}$, respectively, but only ~1.4 for BS, implying that urban BS is more dominated by local sources and less impacted by the air-mass origin than other components of PM_{10}. Consequently there is a lower urban BS:PM_{10} ratio for polluted European trajectories. (Fig. 7 shows BS to be second highest on average for the Arctic trajectories but this is, in part, due to the urban monitoring site being on the south side of the city centre. The key point is the lower variability of BS with air mass source region than for $PM_{2.5}$ and PM_{10}). Rural BS was also elevated for the

European and local circulation air-mass back trajectories, but the differences amounted only to ~1 μg m^{-3} in absolute concentration (data not plotted). Thus, although there exists a "background" component of BS influenced by long-range transport, it is a minor source of BS compared with BS generated in the local vicinity of the urban area, as highlighted above from consideration of urban:rural BS ratios. These analyses have yielded more detailed support to the earlier conclusion of Buchanan *et al*. (2002) that BS is these days rather less influenced by long-range transport than $PM_{2.5}$ and PM_{10}.

Conclusions

European data for collocated $PM_{2.5}$ and PM_{10} remain fairly scarce despite strong moves towards defining $PM_{2.5}$ air quality objectives for protection of human health. The combination of PM and black smoke measurements can provide insight into controlling influences on receptor particle concentrations. This work has shown that $PM_{2.5}$ in background air in a medium-sized UK city is likely to be within potential future air quality objectives, and that, on average, $PM_{2.5}$ is strongly correlated with PM_{10}. If chronic exposure to particles in the long-term (years) dominates the adverse health burden then measurement of one metric in a locality may suffice. On the other hand if short-term (daily) variability in exposure to particles is important then this has been shown to be driven more by variation in $PM_{2.5}$, than by PM_{coarse} or by combustion-related particles as characterised by black smoke. Either way, variation in PM is strongly influenced by local and synoptic meteorology outwith local control measures, with combustion-related dark particles and coarse particles related more to local sources dependent on dispersion and suspension, and finer particles related more to regional scale transport. By understanding the variance of the sources and determinants of the particle matter heterogeneous mix it may be possible to offer a better interpretation of the epidemiologic data associating particulate air pollution with adverse health outcomes.

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Figure Captions

Figure 1: Relationship between daily PM_{10} and $PM_{2.5}$ in urban background air in Edinburgh. (Summer, Apr-Sep; Winter, Oct-Mar).

Figure 2: Distribution of ratios between daily values of PM_{10}, $PM_{2.5}$ and black smoke (BS) in urban background air in Edinburgh. (Summer, Apr-Sep; Winter, Oct-Mar).

Figure 3: Relationship of $PM_{2.5}$:PM_{10} ratio against $PM_{2.5}$ for daily measurements in urban background air in Edinburgh.

Figure 4: Relationship between daily PM_{10} and black smoke and local windspeed in Edinburgh. (Summer, Apr-Sep; Winter, Oct-Mar).

Figure 5: Ratio of PM_{coarse}/BS with windspeed in Edinburgh, and best-fit lines. (Summer, Apr-Sep; Winter, Oct-Mar).

Fig. 6: Wind roses of collocated PM_{10}, $PM_{2.5}$, PM_{coarse} and BS in background air in Edinburgh. Data points are moving averages of three consecutive 10° wind-sector values.

Fig 7: Apportionment of daily PM_{10} and $PM_{2.5}$, and $PM_{2.5}$/PM_{10} ratio, in Edinburgh by geographical sector of 5-day air-mass back-trajectory. Clusters are ordered in descending median PM_{10} value.

Fig. 1: Relationship between daily PM_{10} and $PM_{2.5}$ in urban background air in Edinburgh. (Summer, Apr-Sep; Winter, Oct-Mar).

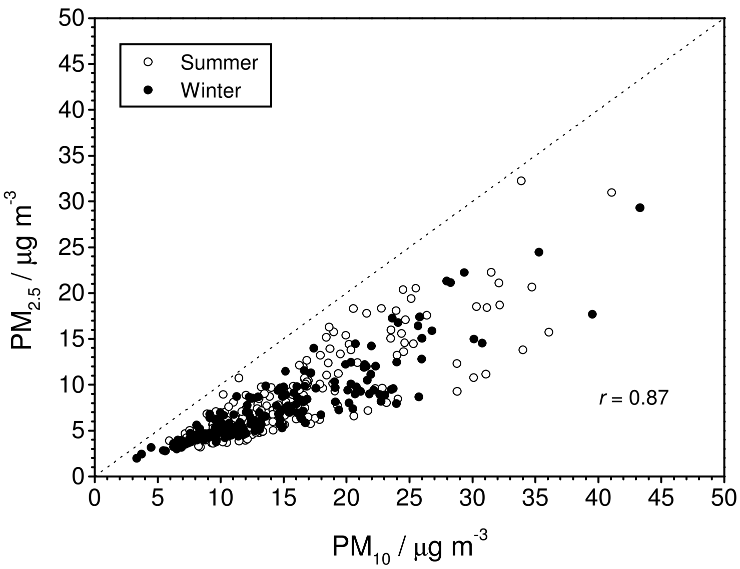


Fig 2: Distribution of ratios between daily values of PM_{10}, $PM_{2.5}$ and black smoke (BS) in urban background air in Edinburgh. (Summer, Apr-Sep; Winter, Oct-Mar).

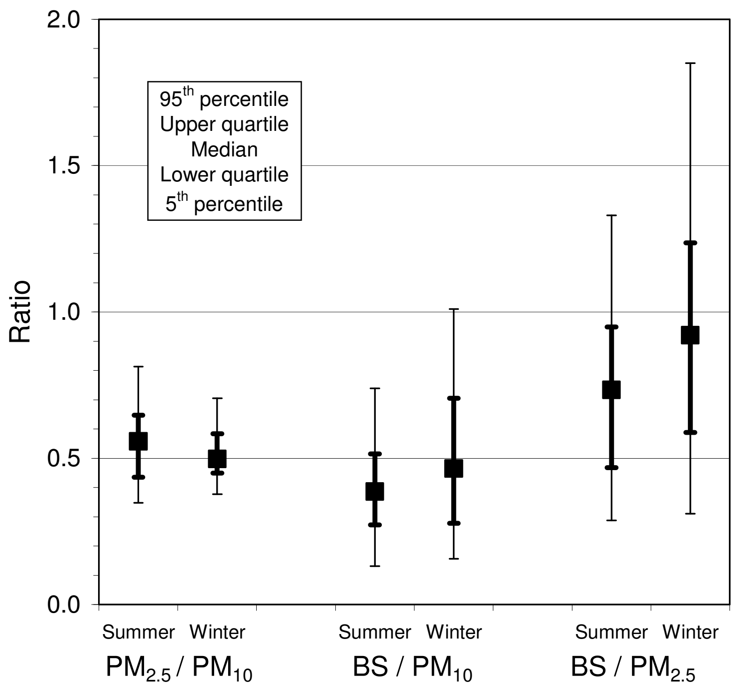


Fig 3: Relationship of $PM_{2.5}$:PM_{10} ratio against $PM_{2.5}$ for daily measurements in urban background air in Edinburgh.

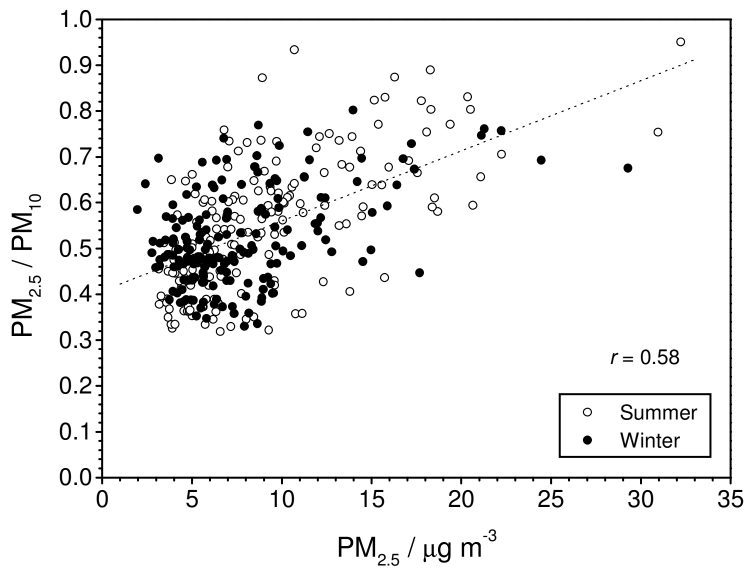


Fig. 4: Relationship between daily PM_{10} and black smoke and local windspeed in Edinburgh. (Summer, Apr-Sep; Winter, Oct-Mar).

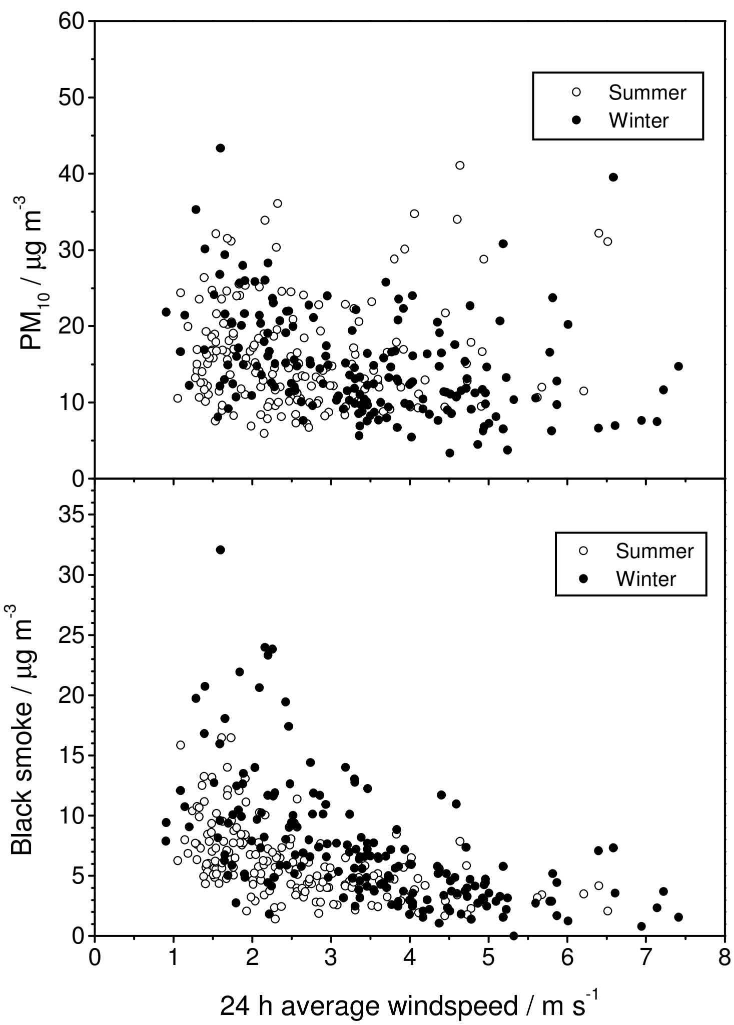


Fig. 5: Ratio of PM_{coarse}/BS with windspeed in Edinburgh, and best-fit lines. (Summer, Apr-Sep; Winter, Oct-Mar).

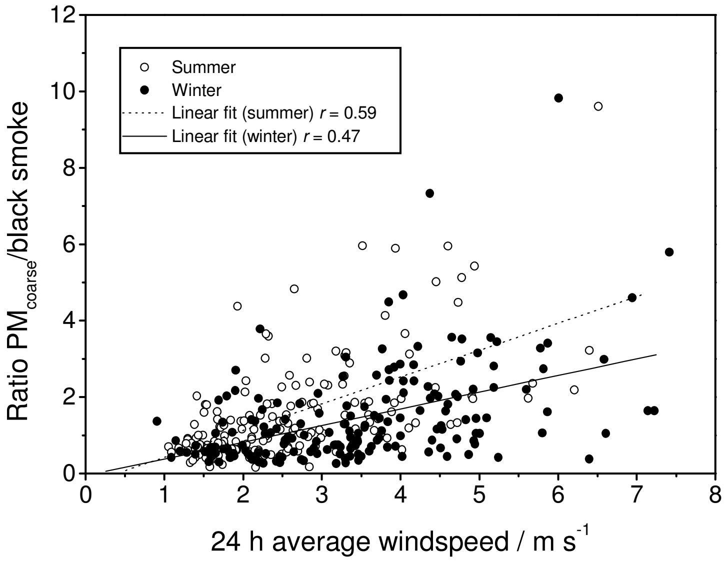


Fig. 6: Wind roses of collocated PM_{10}, $PM_{2.5}$, PM_{coarse} and BS in background air in Edinburgh. Data points are moving averages of three consecutive 10° wind-sector values.

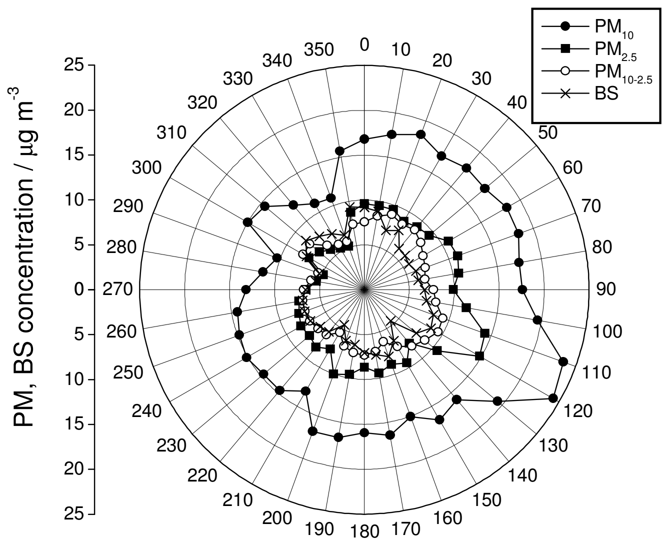
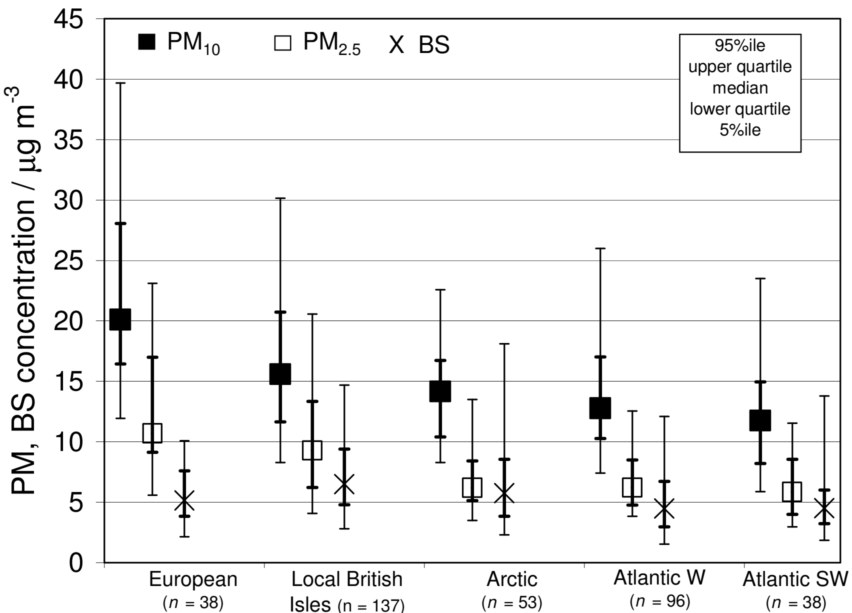


Fig 7: Apportionment of daily PM_{10} and $PM_{2.5}$, and $PM_{2.5}/PM_{10}$ ratio, in Edinburgh by geographical sector of 5-day air-mass back-trajectory. Clusters are ordered in descending median PM_{10} value. The number of trajectories in each cluster is given in the axis labels.



Median daily ratios	European	Local British Isles	Arctic	Atlantic W	Atlantic SW
$PM_{2.5}/PM_{10}$	0.58	0.60	0.47	0.48	0.51
BS/PM_{10}	0.24	0.46	0.48	0.37	0.41